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LCA OF WASTE MANAGEMENT SYSTEMS

Carbon and life cycle implications of thermal recovery from the organic fractions of municipal waste

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Abstract
Purpose The aim of this research was to determine the optimum way of recovering energy from the biodegradable fractions of municipal waste. A part-life cycle study was carried out on the following wastes: paper, food waste, garden waste, wood, non-recyclable mixed municipal waste and refuse-derived fuel. The energy recovery processes considered were incineration, gasification, combustion in dedicated plant, anaerobic digestion and combustion in a cement kiln.
Methods The life cycle assessment (LCA) was carried out using WRATE, an LCA tool designed specifically for waste management studies. Additional information on waste composition, waste collection and the performance of the energy recovery processes was obtained from a number of UK-based sources. The results take account of the energy displaced by the waste to energy processes and also the benefits obtained by the associated recycling of digestates, metals and aggregates as appropriate.
Results and discussion For all the waste types considered the maximum benefits in terms of climate change and non-renewable resource depletion would be achieved by using the waste in a cement kiln as a substitute fuel for coal. When considering the impacts in terms of human toxicity, aquatic ecotoxicity, acidification and eutrophication, direct combustion with energy recovery was the best option. The results were found to be highly sensitive to the efficiency of the energy recovery process and the conventional fuel displaced by the recovered energy.
Conclusions and recommendations This study has demonstrated that LCA can be used to determine the benefits and burdens associated with recovering energy from municipal waste fractions. However, the findings were restricted by the lack of reliable data on the performance of waste gasification and anaerobic digestion systems and on the burdens arising from collecting the wastes. It is recommended that further work is carried out to address these data gaps.

Keywords Energy recovery · Life cycle assessment · Municipal waste · Waste management · WRATE

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1 Introduction

The management of solid wastes has a significant impact on the environment. For example, landfills were estimated to produce 36 % of England's methane emissions in 2010 (Department of Energy and Climate Change 2012). Furthermore, leachate from landfills can also lead to pollution of ground and surface waters. These impacts are recognised in international and national environmental policies such as the EU's revised Waste Framework Directive (European Commission 2008) and England's Waste Review (Defra 2011). Both documents call for the increased recovery of materials and energy from waste and

the latter encourages specific technologies such as the anaerobic digestion (AD) of food waste. However, it is important to establish what evidence there is to underpin such policies. This can only be done if reliable information is available on the environmental impacts and benefits for each stage of each waste management option.

The biodegradable fractions of waste produce methane when landfilled but such wastes are also sources of renewable energy and, if used to replace fossil fuels, will reduce fossil fuel-related greenhouse gas emissions and conserve conventional fuels. Energy recovery processes also recycle some materials through the recovery of digestates, metals and aggregates, leading to additional environmental benefits from reduced energy and resources use. Although materials recycling can have environmental advantages in comparison with thermal recovery (Kärnä et al. 1994 and Dahlbo et al. 2005), there are technical, financial and environmental reasons why it is impossible to recycle all the theoretically recyclable waste. Therefore, thermal recovery needs to be fully assessed as an option.

This paper determines the environmental burdens and benefits from recovering energy from the main, biodegradable municipal waste fractions (paper, food waste, garden waste and wood). In addition, partially biodegradable non-recyclable mixed municipal waste and refuse-derived fuel (RDF) were considered. The RDF was an “idealised product” comprising 95 % of the paper, card, dense plastic, textiles and wood from the non-recyclable residual waste (Table 1). It is recognised that RDF would, in reality, contain some food and garden wastes and other contaminants, but the idealised RDF with a lower heating value of 16.8 MJ kg^{-1} and RDF yield of 34 % of the residual waste input compares well with published RDF heating value (12–20 MJ kg^{-1}) and yield (23–50 %) data (European Commission 2003).

The thermal recovery processes are incineration or energy from waste (EfW) (for all fractions except RDF), gasification (paper, wood and RDF), combustion in dedicated plant (wood and RDF), anaerobic digestion (food and garden wastes) and combustion in a cement kiln (all materials except mixed residual waste).

Table 1 Idealised RDF composition

Component	Value
Paper	32 %
Card	13 %
Dense plastic	36 %
Textiles	10 %
Wood	9 %
Lower heating value	16.8 MJ kg^{-1}
Yield	34 % of original residual waste

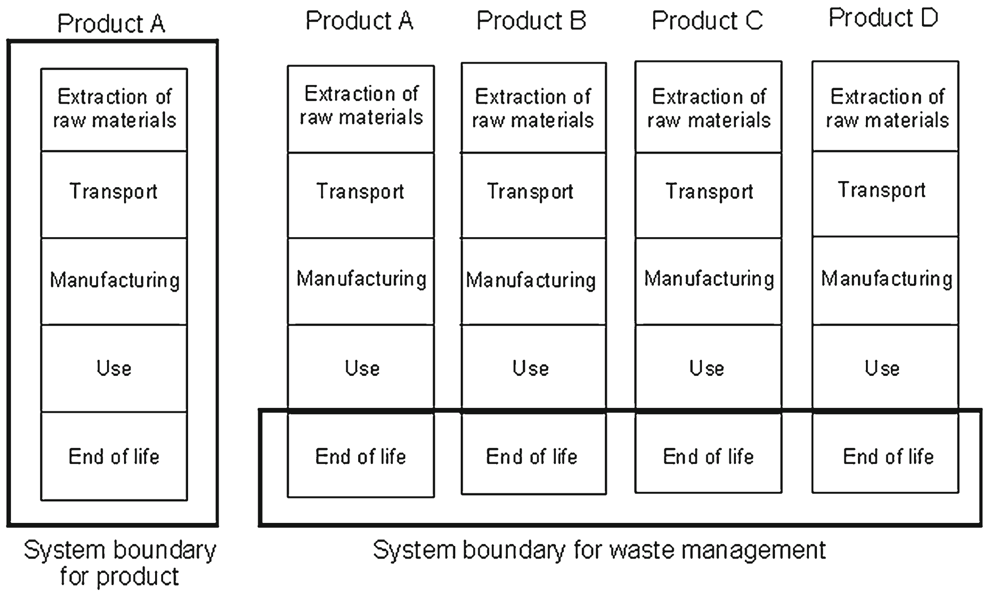
As a precursor to this research, an assessment of the energy balance of each waste was carried out and has been reported elsewhere (Burnley et al. 2011). That analysis provided some information on the merits of the different waste treatment options, but other environmental impacts (overall global warming potential, acidification, eutrophication, human and aquatic toxicity and resource depletion) and those arising from the recycling of materials and landfilling of residues were not considered. This research uses life cycle assessment (LCA) to calculate the impacts and benefits of the whole waste management system including any environmental benefits resulting from the use of waste in place of other resources. As far as possible, the requirements of the ISO 14040 series (BS EN ISO 14040 2006) were followed.

Waste and Resources Assessment Tool for the Environment (WRATE) is a life cycle assessment tool developed by the Environment Agency (EA) for England and Wales. WRATE uses peer-reviewed data on the capital and operating burdens of conventional and novel waste management options obtained by the EA from discussions with plant operators, equipment suppliers and the literature. WRATE also includes data on related operations such as waste transport and storage (Environment Agency 2010). Conventional LCA tools evaluate the environmental burdens of a particular product from raw material extraction, product manufacture, distribution and use, through to product disposal. WRATE and other waste management LCA tools (Boldrin et al. 2011) consider the entire waste stream from the moment waste is discarded through collection, processing and final disposal of any residues to landfill. These tools also take account of any environmental benefits from the recycling of materials, compost and digestate and the recovery of energy. The relationship between waste management and conventional LCA systems is shown in Fig. 1 (McDougall et al. 2001).

2 Review of previous studies

Bates (2009) used WRATE to carry out an LCA of the management of residual municipal waste. Energy from waste incineration (power only and combined heat and power (CHP)), biodrying to produce refuse-derived fuel for direct combustion and mechanical biological treatment to produce a fuel fraction and an organic-rich fraction for composting or anaerobic digestion were all compared with landfill. In terms of climate change, direct combustion and biodrying to produce RDF were the only options that showed an overall benefit with the remaining processes leading to a net increase in climate change emissions. The results were found to be highly sensitive to the source of the power displaced with the overall benefits decreasing with the move from the baseline (50 % coal and 50 % gas)

Fig. 1 Relationship between waste management and conventional LCAs (McDougall et al. 2001)



through gas only to a "low carbon" source (hydropower and gas).

Watson et al. (2009) used WRATE in an LCA of the combustion and gasification of municipal waste. The results concentrated on the climate change impacts and concluded that, regardless of the technology selected, CHP provided the best option. For a given energy conversion efficiency, there appeared to be little difference between the combustion and gasification processes. However, it should be noted that the gasification process was based on a model of a downdraft gasifier operating at 800°C produced by Fock et al. (2000). Fock et al. developed this model for biomass and Watson et al. assumed that raw municipal waste can be gasified in the same process as biomass without the need to produce a homogeneous refuse-derived fuel prior to gasification. This is in contrast to the models used by Burnley et al. (2011) where RDF production was a necessary prior to gasification.

Kranet et al. (2010) determined the CO_2 impacts of burning various types of garden waste with energy recovery. The CO_2 reductions ranged from 1,040 kg per tonne burned for high calorific value, woody material to 126 kg of CO_2 per tonne for herbaceous/grassy waste. However, composting the waste and using it as a peat substitute proved to be more beneficial; largely due to the reduction in CO_2 emissions from the degradation of peat which occurs once it is extracted from the peat beds.

Michaud et al. (2010) published an update of a previous review of LCA studies on the environmental benefits of recycling which also considered some energy recovery comparisons. For food and garden waste, anaerobic digestion appeared to be the best option but, for wood and paper, the balance between energy recovery by incineration and materials recycling was not clear cut. In the case of paper, the climate change impacts of recycling are dependent on the assumptions made about the CO_2 emissions from the production of the energy used when making paper from recycled feedstock and virgin resources. Similarly, the energy recovery climate change impacts depend on the fuel mix used to generate the power substituted by the waste-derived power. For wood processing, the better option depends on the recycling process and on the efficiency of the energy recovery process.

3 Materials and methods

3.1 Goal and scope

The goal, scope, functional unit and target audience of this study are summarised in Table 2. The sources of data referring to the composition and properties of the waste and the performance of the waste collection are given below. Other environmental burdens were calculated by WRATE using the Ecoinvent LCA database version 2.01 (Frischknecht et al. 2005).

The system under consideration is shown in Fig. 2. As shown in Fig. 1, the manufacture and use of the materials that become wastes are not considered. Therefore, all waste is assumed to start with zero environmental burdens and only the emissions from, the resources used by and any consequent resource savings by, the waste management system are quantified in this LCA.

3.2 Inventory analysis

The inventories were calculated using data from a number of sources. For conventional waste management processes

Table 2 Basis of the study

Goal	To quantify the environmental burdens associated with the management of organic household waste materials by conventional incineration, combustion in dedicated plant, gasification and anaerobic digestion as appropriate. The wastes covered are food waste, garden waste, waste paper, waste wood municipal waste (after removal of recyclable materials by kerbside collection schemes) and refuse-derived fuel manufactured from municipal waste.
Functional unit	The management of 1 tonne of each waste material.
Scope	The system covered begins at the point that the waste is set out by the householders. Collection burdens over and above those associated with the normal non-recyclable waste collection are included. Burdens associated with waste preparation and treatment and the landfill of restudies are also included. It is assumed that all thermal processes are used to generate electrical power and the offset burdens from this are included. Compost, digestate and other materials reclaimed for the waste are recycled with the displacement of an equivalent mass of material produced from virgin resources.
Target audience	This study is intended to inform waste management professionals in policymaking, regulatory, local authority, industrial and academic roles.

such as energy from waste and landfill, the default peer-reviewed data from WRATE were used. For gasification, anaerobic digestion and RDF processes, where full-scale operating plant are either limited in number or not currently operational in the UK, data on plant performance and resource consumption were taken from the literature, published pilot-scale trials or from modelling studies (Consonni et al. 2005; Banks et al. 2011; Burnley et al. 2011). The capital burdens of these facilities were taken from WRATE and based on similar facilities. However, it should be noted

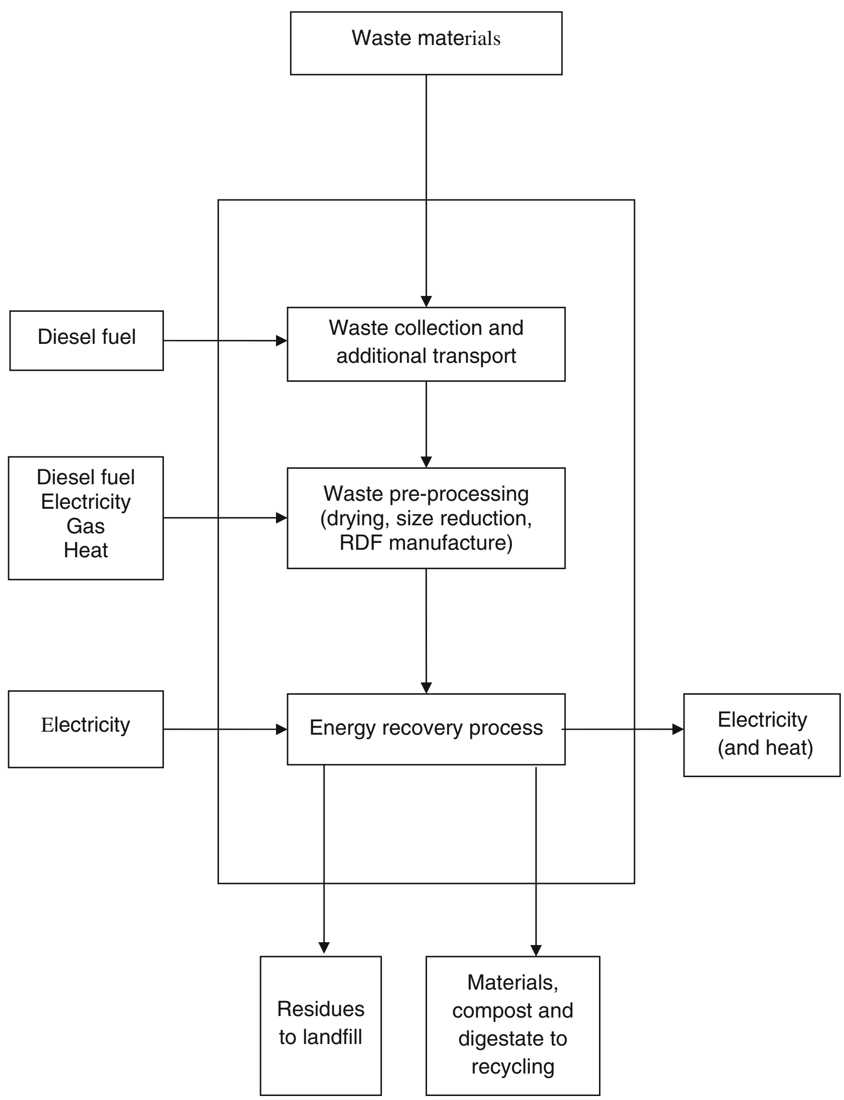


Fig. 2 System boundaries

that these burdens do not contribute significantly to the overall burdens, so any errors caused by the use of such data will be small.

It proved difficult to obtain reliable data on the fuel consumption and emissions from refuse collection vehicles. The data used were obtained from the literature and from personal contacts within the waste management industry; for details see Burnley et al. (2011). Capital burdens from waste collection were based on data supplied by the vehicle manufacturers, which had been peer-reviewed and included in WRATE. In this study, we only considered the marginal waste collection impacts which arise when an additional, separate collection is required over and above the normal waste collection which would take place regardless of the waste management process used. This applied to the collections of food, garden waste and paper. The introduction of separate collect schemes has the potential to result in a reduction of the emissions from collecting the remaining residual waste. However, the impact of these savings is likely to be small and is not considered in this analysis.

3.3 Impact assessment

The environmental burdens calculated by WRATE were categorised and then characterised using the Ecoinvent database to calculate the environmental impacts. The categories are a sub-set of the CML 2001 (Guinée 2002) categories considered by the UK's Department for Environment Food and Rural Affairs (Defra) to be most relevant for this study.

Global warming potential over 100 years expressed as kg CO_2 equivalent;
Acidification potential expressed as kg SO_2 equivalent;
Generic eutrophication potential expressed as kg PO_4 equivalent;
Freshwater aquatic ecotoxicity (FAETP infinite) expressed as kg 1,4-dichlorobenzene equivalent;
Human health (HTP infinite) expressed as kg 1,4-dichlorobenzene equivalent;
Depletion of abiotic resources expressed as kg antimony equivalent.

The results for the six wastes are presented in Tables 3, 4, 5, 6, 7 and 8. It should be noted that the results are highly dependent on the characteristics of the recovery plant. This is particularly true for the energy recovery options where the source of the conventional fuel displaced also has important implications. This is discussed in the following section, but in all instances it is assumed that the power displaced is derived from the UK's non-nuclear and non-renewable

Table 3 Waste paper management environmental burdens (per tonne of waste managed)

	AD					EfW					Gasification					Cement				
	Total	Transport	Recycling	Process	Landfill	Total	Transport	Recycling	Process	Landfill	Total	Transport	Recycling	Process	Landfill	Total	Transport	Recycling	Process	Landfill
Climate change (kg CO_2-Eq)	−109	40	−57	−92	0	−482	1	0	−483	0.4	−376	36	0	−461	0.3	−1,578	79	0	−1,658	0
Human toxicity (kg 1,4-DCB-Eq)	100	15	92	−6	0	−37	0.3	0	−42	5	−26	14	0	−46	4	27	25	0	1	0
Aquatic eco-toxicity (kg 1,4-DCB-Eq)	14	3	12	−0.4	0	−5	0.07	0	−7	2	−2	3	0	−7	2	5	5	0	0	0
Acidification (kg SO_2-Eq)	0.6	0.2	−0.2	0.5	0	−0.2	0.01	0	−0.2	0.002	−0.09	0.2	0	−0.3	0.002	0.4	0.4	0	0	0
Eutrophication (kg PO4-Eq)	0.4	0.04	0.3	0.1	0	0.05	0.001	0	0.05	0.001	0.02	0.04	0	−0.02	0.001	0.4	0.07	0	0.3	0
Resource depletion (kg Sb-Eq)	−0.6	0.3	−0.1	−0.8	0	−4	0.09	0	−4	0.01	−3	0.3	0	−4	0.01	−10	0.7	0	0	0

Table 4 Waste food management environmental burdens (per tonne of waste managed)

	AD					EfW					Cement				
	Total	Transport	Recycling	Process	Landfill	Total	Transport	Recycling	Process	Landfill	Total	Transport	Recycling	Process	Landfill
Climate change (kg CO_2-Eq)	−183	38	−57	−165	0	−125	1	0	−126	0.4	−447	78	0	−525	0
Human toxicity (kg 1,4-DCB-Eq)	91	15	91	−15	0	5	0.4	0	−0.8	6	29	24	0	5	0
Aquatic eco-toxicity (kg 1,4-DCB-Eq)	13	3	12	−2	0	2	0.07	0	−0.06	2	6	5	0	0.7	0
Acidification (kg SO_2-Eq)	0.8	0.2	−0. 2	0.8	0	0.3	0.005	0	0.3	0.003	0.6	0.4	0	0.2	0
Eutrophication (kg PO_4-Eq)	0.5	0.04	0.3	0.1	0	0.1	0.001	0	0.1	0.001	0.2	0.07	0	0.1	0
Resource depletion (kg Sb-Eq)	−1	0.3	−0.1	−1	0	−1.2	0.009	0	−1	0.008	−3	0.7	0	−4	0

Table 5 Garden waste management environmental burdens (per tonne of waste managed)

	AD					EfW					Cement				
	Total	Transport	Recycling	Process	Landfill	Total	Transport	Recycling	Process	Landfill	Total	Transport	Recycling	Process	Landfill
Climate change (kg CO_2-Eq)	−147	38	−57	−129	0	−161	1	0	−162	0.4	−563	73	0	−636	0
Human toxicity (kg 1,4-DCB-Eq)	95	15	91	−11	0	1	0.4	0	−5	6	30	24	0	6	0
Aquatic eco-toxicity (kg 1,4-DCB-Eq)	14	3	12	−1	0	1	0.07	0	−0.8	2	6	5	0	0.8	0
Acidification (kg SO_2-Eq)	0.7	0.2	−0.2	0.7	0	0.3	0.005	0	0.3	0.003	0.5	0.4	0	0.2	0
Eutrophication (kg PO_4-Eq)	0.5	0.04	0.3	0.1	0	0.1	0.001	0	0.1	0.001	0.2	0.07	0	0.1	0
Resource depletion (kg Sb-Eq)	−0.9	−0.1	−0.1	−1	0	−1	0.009	0	−1	0.008	−4	0.6	0	−4	0

Table 6 Wood waste management environmental burdens (per tonne of waste managed)

	EfW				Dedicated combustion				Gasification				Cement			
	Total	Transport	Process	Landfill	Total	Transport	Process	Landfill	Total	Transport	Process	Landfill	Total	Transport	Process	Landfill
Climate change (kg CO_2-Eq)	−758	0.5	−758	0.1	−1,180	13	−1,190	0.1	−683	12	−696	0.07	−2,500	44	−2,544	0
Human toxicity (kg 1,4-DCB-Eq)	−71	0.2	−75	3	−118	4	−125	3	−69	4	−74	0.9	10	11	−1	0
Aquatic eco-toxicity (kg 1,4-DCB-Eq)	−11	0.03	−13	1	−20	0.8	−21	0.999	−11	0.8	−12	0.3	0.6	2	−2	0
Acidification (kg SO_2-Eq)	−0.5	0.002	−0.5	0.001	−1	0.06	−1	0.001	−0.6	0.06	−0.7	0.0005	0.04	0.2	−0.2	0
Eutrophication (kg PO_4-Eq)	0.03	0.0004	0.03	0.001	−0.03	0.02	−0.04	0.001	−0.04	0.01	−0.05	0.0002	0.6	0.04	0.6	0
Resource depletion (kg Sb-Eq)	−6	0.004	−6	0.003	−10	0.1	−10	0.003	−6	0.1	−0.6	0.002	−16	0.4	−17	0

average (coal 46.6 %, gas 49.3 % and oil 3.3 %). Similarly, the Ecoinvent database used by WRATE assumes that the power saved by recycling paper and metals is derived from the European average.

4 Interpretation and discussion

4.1 Paper

Recovering energy from waste paper by conventional incineration and gasification shows an overall benefit in all categories except eutrophication where there is a small disbenefit. Although the engine used to generate the power in the gasification process is more efficient than the incineration steam cycle, this advantage is negated by the low solid to gas conversion efficiency of the gasifier, so overall, incineration produces the greater benefits (although the gasifier is slightly better in terms of eutrophication due to the lower NO_x emissions).

Paper contains a high proportion of renewable carbon, although much of this is present in the form of lignocellulose which degrades only very slowly. Therefore, the digestion of paper produces a relatively low biogas yield and power output, so the benefits in terms of climate change are much lower than for the thermal processes. AD gives overall disbenefits in terms of human and aquatic toxicity due to the deposition of heavy metals in the soil, eutrophication from nitrate addition to water and acidification arising from the additional transport burdens. It is recognised that paper would only be treated by AD along with other wastes to ensure a realistic carbon/nitrogen ratio. However, this system was included for purposes of comparison.

Very little information is published on the performance of AD systems with municipal wastes, so further research is necessary to confirm the gas yields and hence power output. This is also true for the cases of food waste and garden waste digestion.

The cement manufacture option shows the greatest benefits in terms of climate change and resource depletion (see Table 3). There are two reasons for this. Firstly, the waste is converted to useful energy at the same efficiency as the fuel that is displaced. This is not the case when converting wastes to power (for example a conventional energy from waste plant will be around 10 percentage points less efficient than a conventional power plant). Secondly, cement manufacture displaces high carbon coal rather than a lower carbon coal/gas mix as explained above. However, disbenefits arise in all other categories from transporting the waste to the cement kiln in the case of human and aquatic toxicity and acidification (an assumed distance of 200 km) and from the NO_x emissions of the kiln in the case of

Table 7 Residual municipal waste management environmental burdens (per tonne of waste managed)

	Conventional EfW					High-efficiency EfW				
	Total	Transport	Recycling	Process	Landfill	Total	Transport	Recycling	Process	Landfill
Climate change (kg CO_2-Eq)	−123	3	−89	−37	0.1	−267	3	−89	−180	0.1
Human toxicity (kg 1,4-DCB-Eq)	−336	0.8	−314	−27	4	−351	0.8	−314	−42	4
Aquatic eco-toxicity (kg 1,4-DCB-Eq)	−30	0.2	−27	−5	1	−33	0.21	−27	−7	1
Acidification (kg SO_2-Eq)	−0.3	0.01	−0.4	0.04	0.0008	−0.5	0.01	−0.4	−0.1	0.0008
Eutrophication (kg PO_4-Eq)	0.05	0.003	−0.04	0.08	0.0007	0.03	0.003	−0.04	0.06	0.0007
Resource depletion (kg Sb-Eq)	−4	0.02	−0.6	−3	0.002	−5	0.02	−0.6	−4	0.002

eutrophication. Many of these disbenefits would be eliminated if the waste was produced close to the kiln.

This analysis suggests that cement manufacture is the best option in terms of climate change and resource depletion and conventional incineration is the best option for all other categories.

4.2 Food waste

Food waste has a high moisture content (around 63 % (Department of the Environment 1994)); consequently, the energy recovered in all three processes is very low (resulting in CO_2 reductions of 125, 183 and 447 kg t^{-1} for EfW, digestion and cement manufacture respectively), although the AD energy output is greater than for the AD of paper (a CO_2 saving of 109 kg t^{-1}) due to the low biodegradability of paper over the time period spent in the digester.

All three processes considered have an overall benefit in terms of climate change and resource depletion, but disbenefits in the other categories. As well as fossil fuel replacement, AD has an additional climate change benefit related to the displacement of nitrogen-based fertilisers, assuming that the digestate is applied to land as a fertiliser/soil improver.

The main environmental burdens for AD arise from the application of heavy metals to land when the digestate is used (human and aquatic toxicity), ammonia and NO_x emissions from the digestion and combustion processes (acidification) and nitrates discharged to water when the digestate is used (eutrophication). In the case of incineration, landfill of the ashes generates the greatest contribution to human and aquatic toxicity and NO_x emissions contribute most to eutrophication and acidification.

In summary, cement production is the best option in terms of climate change and resource depletion while conventional EfW has the smallest burdens in the other categories. AD performs better than EfW in terms of climate change but, if the digestate is not used for any beneficial purpose, AD is no better than EfW in this respect.

4.3 Garden waste

Garden waste management shows a similar picture to food waste with overall benefits for cement production, incineration and AD processes in terms of climate change and resource depletion and overall burdens in the other categories.

As with paper and food waste, the benefits and burdens of AD depend on the fate of the digestate. If the digestate is applied to land the climate change and resource use benefits are increased, but the other categories all result in a disbenefit.

The combustion-related results can be compared with those of Kranet et al. (2010). For herbaceous garden waste, Kranert et al. reported an overall climate change benefit of 126 kg CO_2-Eq per tonne burned. This compares well with our results given that Kranert et al. did not specify the assumed efficiency of their combustion process or the displaced fossil fuel mix.

Overall, this analysis suggests that combustion in a cement kiln has the greatest climate change and resource use benefits and EfW performs best in the other categories.

4.4 Wood waste

Combustion in a cement kiln shows the greatest climate change and resource use benefits with the high-efficiency dedicated combustor shows the greatest benefits in all the other categories. With the exception of eutrophication (due to NO_x emissions), energy recovery through conventional incineration is more environmentally beneficial than gasification due to the lower conversion efficiency of the gasifier as is the case with paper gasification.

The climate change results for the dedicated combustion process of 1,180 kg CO_2-Eq per tonne compare well with Kranet et al. (2010) findings for “woody” waste of 1,040 kg CO_2-Eq per tonne.

4.5 Residual municipal waste

Managing municipal waste through energy recovery realises environmental benefits in all categories except eutrophication (due to the NO_x emissions from combustion).

Table 8 Refuse-derived fuel management environmental burdens (per tonne of waste managed)

	Dedicated combustion					Gasification					Cement				
	Total	Transport	Recycling	Process	Landfill	Total	Transport	Recycling	Process	Landfill	Total	Transport	Recycling	Process	Landfill
Climate change (kg CO_2-Eq)	−37	7	−110	−12	78	−47	7	−110	−22	78	−589	21	−110	−579	78
Human toxicity (kg 1,4-DCB-Eq)	−393	2	−409	13	0.06	−388	2	−409	19	−0.4	−362	6	−409	42	−1
Aquatic eco-toxicity (kg 1,4-DCB-Eq)	−37	0.4	−35	−3.67	1	−37	0.4	−35	−3	1	−32	1	−35	0.8	0.7
Acidification (kg SO_2-Eq)	−0.6	0.04	−0.5	−0.2	0.004	−0.6	0.04	−0.5	−0.1	0.003	−0.3	0.1	−0.5	0.05	0.002
Eutrophication (kg PO_4-Eq)	0.2	0.007	−0.05	0.03	0.2	0.1	0.007	−0.05	0.01	0.2	0.4	0.02	−0.05	0.2	0.2
Resource depletion (kg Sb-Eq)	−3	0.06	−0.7	−2	−0.3	−3	0.06	−0.7	−2	−0.3	−6	0.2	−0.8	−6	−0.3

Table 7 demonstrates the importance of thermal efficiency on the environmental impacts of thermal treatment of wastes. Moving from the current 21 % net efficiency plant to the proposed high-efficiency (28 %) incineration plant doubles the climate change-related benefits and improves all other categories.

Bates (2009) considered the climate change impacts of mass burn EfW of residual waste making similar assumptions about the displaced fossil fuel and produced similar overall results. However, this hides some important difference. Bates assumed that 30 % of the non-ferrous metal in the bottom ash was reclaimed compared with 70 % in this case and the overall thermal efficiency was taken as 23 % compared with 21 % in the current study. Repeating our analysis using Bates' values for these factors resulted in an overall climate change impact of −131 kg CO_2-Eq still showing excellent agreement with Bates' value of −123 kg CO_2-Eq.

The materials recycling associated with the incineration process has a significant impact (Table 9). For the conventional 21 % thermal efficiency plant, recycling is responsible for the bulk of the benefits in climate change, human and aquatic toxicity and acidification. This reflects both the fact that around 40 % of the CO_2 released on burning the residual waste is from fossil sources and also the point discussed above that the efficiency of the EfW plant is lower than that of the fossil fuel output that is displaced. A detailed analysis of this result using WRATE showed that these benefits accrued from the recycling of aluminium and, to a lesser extent, ferrous metals from the ash. The Ecoinvent LCA database entry for aluminium assumes 9 kg of fossil CO_2 is saved per kilogram of aluminium recycled. This is based on average European data for electricity consumption by the aluminium production process. If this was to be replaced by aluminium manufactured using hydropower or other low-carbon energy sources, the benefits from recycling aluminium would be greatly reduced.

4.6 Refuse-derived fuel

Manufacturing refuse-derived fuel may have economic or logistical benefits in areas where the quantity of waste produced does not justify a mass burn incinerator. However, one tonne of municipal waste produces only 340 kg of RDF with the remainder being sent to landfill apart from a small amount of recycling. Consequently, the dual impact of a lower power output and increased biodegradable waste being sent to landfill, makes RDF less environmentally attractive than mass burn incineration.

All three RDF-based options considered (see Table 8) show an overall environmental benefit in each category with the exception of eutrophication, again due primarily to NO_x emissions. These benefits are due to the recovery of aluminium (and to a lesser extent steel and glass) for recycling

Table 9 Breakdown of residual waste EfW benefits and impacts

	Conventional EfW					High-efficiency EfW				
	Total	Transport	Recycling	Process	Landfill	Total	Transport	Recycling	Process	Landfill
Climate change (kg CO_2-Eq)	−123	3	−89	−37	0.1	−267	3	−89	−180	0.1
Human toxicity (kg 1,4-DCB-Eq)	−336	0.8	−314	−27	4	−351	0.8	−314	−42	4
Aquatic eco-toxicity (kg 1,4-DCB-Eq)	−30	0.2	−27	−5	1	−33	0.21	−27	−7	1
Acidification (kg SO_2-Eq)	−0.3	0.01	−0.4	0.04	0.0008	−0.5	0.01	−0.4	−0.1	0.0008
Eutrophication (kg PO_4-Eq)	0.05	0.003	−0.04	0.08	0.0007	0.03	0.003	−0.04	0.06	0.0007
Resource depletion (kg Sb-Eq)	−4	0.02	−0.6	−3	0.002	−5	0.02	−0.6	−4	0.002

Note that process benefits/burdens represent the savings from fossil fuel displacement less the burdens associated with burning the residual waste

during the RDF production process (Table 10). If these benefits were not included, dedicated combustion and gasification would lead to a net increase in all categories of emissions due to the burdens associated with landfilling the rejected material from the fuel production process.

Overall, combustion in the cement kiln is the best option for RDF regarding climate change and resource depletion. For the remaining categories, there is little difference between gasification and dedicated combustion although gasification gives a slightly better climate change benefit.

The climate change results for RDF combustion are in general agreement with those of Bates in that direct combustion of the raw refuse is by far the better option in both studies. However, Bates' results for RDF demonstrate barely an overall benefit. This difference presents further evidence of the sensitivity of such studies to the operating parameters selected; in this case the proportion of refuse entering the RDF stream, the efficiency of the combustion process and the recovery of materials for recycling (aluminium in particular).

For the gasification of RDF with combustion of the gas in a gas engine, Watson et al. (2009) obtained a climate change figure of −34 kg CO_2–Eq per tonne of waste processed which is similar to the value of −47 kg CO_2-Eq obtained in this study. As is the case in the above comparison with Bates' results, these similar overall figures hide a number of differences which arose due to the different assumptions made about the gasifier performance, power generation efficiency and associated recycling.

5 Overall discussion

This study has assessed for England for the first time the life cycle environmental burdens associated with managing biodegradable municipal wastes. The sources of the principal environmental burdens and benefits for each option are identified and areas where improvements can be made indicated.

For the options dealing with paper, food waste and garden waste where separate collection was required, the impacts associated with collecting the waste were significant. For example, for food waste digestion, the collection element accounts for 15 % of the human toxicity burdens and 25 % of the aquatic eco-toxicity and acidification burdens. Further work to establish the feasibility and environmental burdens associated with combining collections of these wastes with the residual waste using split-container vehicles would be advantageous.

Table 10 RDF manufacture and combustion in dedicated facilities—significance of recycling (per tonne of waste managed)

	Mass recovered (kg)	Climate change (kg CO_2-Eq)	Human toxicity (kg 1,4-DCB-Eq)	Aquatic eco-toxicity (kg 1,4-DCB-Eq)	Acidification (kg SO_2-Eq)	Eutrophication (kg PO_4-Eq)	Resource depletion (kg Sb-Eq)
Recycling total	100	−110	−409	−35	−0.5	−0.05	−0.7
Aluminium	7.1	−76	−398	−34	−0.4	−0.03	−0.4
Ferrous metal	20.8	−34	−10	−0.9	−0.1	−0.01	−0.3
Glass	43	−1	−0.3	−0.07	−0.005	−0.001	−0.1
Aggregate from ash	29	0.5	0.1	0.05	0.002	0.0004	0.005

The metal content of the ash is much lower than in the case of raw refuse, so it is assumed that metal recovery is not carried out as part of the ash recycling process

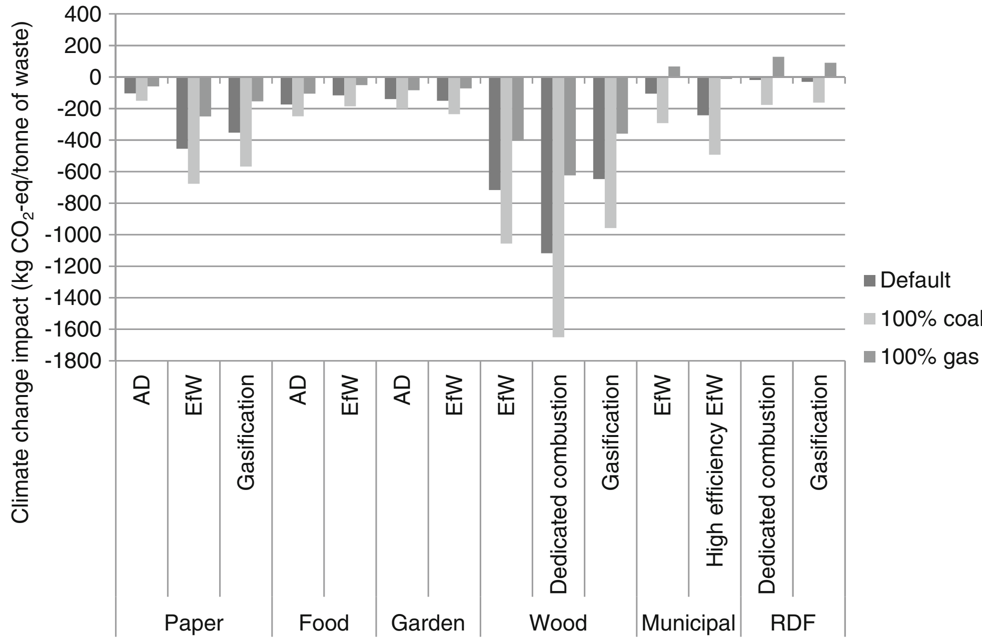


Fig. 3 Sensitivity of residual waste combustion to fuels displaced

For the cement kiln option, the transport of the waste to the kiln is highly significant and accounts for almost all of the environmental burdens. However, the large benefits resulting from reduction in the mining, importing and combustion of coal mean that, with the exception of aquatic ecotoxicity (from transport-related NO_x emissions), the benefits greatly exceed the negative aspects of the process.

The thermal output from all the processes and feedstocks considered could be used to provide process and/or space heating with a reduced or zero electrical power output. Such combined heat and power applications have a potential thermal efficiency of greater than 80 % which would result in an overall improvement in the life cycle impacts for all the wastes and all the options. However, CHP requires a constant demand for the heat energy throughout the year which is dependent on the local conditions rather than on the waste processing technology. For example, the Carbon Trust (2008) suggests that the demand for heat ranges from 20 % of the time for offices to 60 % for process uses. Therefore, to allow comparisons to be made on an equivalent basis, this analysis was restricted to power generation where there is a ready market for the power under all circumstances.

Table 11 Research and information needs

Area	Research needs
Waste collection	Energy consumptions and emissions from collecting mixed waste and separate fractions
	Potential for reducing collection fuel use by integrating collections using split vehicles, trailers etc.
Waste composition	Proximate analysis of waste fractions
	Ultimate analysis of waste fractions
Anaerobic digestion	Gas yield and composition from full-scale plant for a range of feedstocks
	Plant heat and power consumption
	Long-term operating experience
Energy from waste incineration	Reliability of high thermal efficiency plants
Gasification	Gas yield and composition from full-scale plant for a range of feedstocks
	Plant heat and power consumption
	Long-term operating experience
Refuse-derived fuel	Energy consumption of production plants

In all cases where electrical power is generated it is assumed that the power displaced is generated from a mix of coal (46.6 %), natural gas (49.3 %) and oil (3.3 %). These proportions reflect the relative use of the three fuels in the UK excluding nuclear, renewable and imported power. However, it could be argued that the substitution should be 100 % gas (the most marginal fuel due to its high cost on a thermal basis and the ease of reducing its use) or 100 % coal (the highest carbon fuel which would be the first to be reduced if national policy was to reduced carbon emissions). The sensitivity to this is shown in Fig. 3. Of particular note are RDF combustion and gasification and conventional EfW for residual municipal waste where these become a net climate change contributor when substituting gas-derived power.

6 Limitations of the study

Several limitations were identified during the study and these were principally related to a shortage of reliable data

in several areas. These limitations are considered in the following paragraphs.

The data on the chemical composition of waste are over 15 years old (Department of the Environment 1994). Changes in product composition for example, following implementation of the packaging directive (European Commission 1994) may have led to lower heavy metal levels in waste which will influence the environmental burdens related to human and aquatic toxicity.

The values used for the fuel consumed and emissions relating to the additional collection of wastes are based on a very limited sample of one private company and two English local authorities (Burnley et al. 2011). These impacts are significant for paper, food and garden waste collections. Additional data on the collection of these materials in dedicated collections and as part of an integrated collection with other materials would give more confidence in these results.

The treatment of fractions of municipal waste by AD technologies is not widely carried out in the UK. Furthermore, reviews of practice elsewhere in Europe taken from the grey literature generally lack reliable mass and energy balances. This study made use of one demonstration scheme treating food waste (Banks et al. 2011) in combination with laboratory-scale investigations (Godley et al. 2009).

Similarly, there are few reliable data on gasification systems operating on waste materials. Consequently, the yield and composition of the gasifier products was determined by a theoretical modelling exercise (Burnley et al. 2011). The capital burdens and emissions from the gasification system were taken from WRATE's database of generic gasification processes. Deviations in the yield could have significant impacts of the gasification LCA.

7 Conclusions and recommendations

For climate change and resource depletion impacts, the best option for dealing with each waste is to use it as a replacement for coal in a cement kiln. This is because coal is replaced on a 1:1 basis in terms of calorific value. If the use of cement kilns is discounted, for climate change, the best option assessed for each waste is:

- Paper waste—incineration in conventional EfW plant
- Food waste—AD
- Garden waste—incineration in conventional EfW plant
- Wood waste—combustion in dedicated high-efficiency plant
- RDF—gasification
- Residual municipal waste—incineration in high-efficiency EfW plant

For the remaining life cycle impacts assessed, EfW was the best option for all feedstocks except wood, where dedicated combustion was best and RDF, where gasification and EfW performed equally well.

It should be noted that high-efficiency EfW was only considered for residual municipal waste and may outperform the above options if considered for other wastes. The climate change results are also highly sensitive to a number of factors: particularly the fuel mix used to generate the electrical power that is displaced, the methane yield and the fate of the digestate for the AD systems and the distance travelled to the plant, which tends to be further for the cement kiln.

This study has also suffered from a lack of reliable data on the burdens associated with waste collection, the composition of municipal waste and overall plant performance. Further research is required in all these areas as summarised in Table 11.

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